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Tillage and straw management affect PM10 emission potential in subarctic Alaska

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ABSTRACT

Emission of windblown dust and PM10 (particulates $\leq$10 μm in diameter regulated by many nations as an air pollutant) from agricultural soils can impact regional air quality. Little information exists that describes the potential for PM10 and airborne dust emissions from subarctic soils or agricultural soils subject to different tillage and residue management. This study documents the influence of tillage and residue management on fine particulate emissions from a subarctic soil in Interior Alaska. Surface characteristics and properties of a silt loam soil subject to conventional, spring disk, autumn chisel plow, and no tillage, with stubble and loose straw retained or removed from the soil surface after harvest of continuous barley (*Hordeum vulgare* L.), was measured twenty years after establishing treatments. These soil characteristics and properties were used to simulate PM10 emission potential and soil loss from tillage and residue treatments using the Single-event Wind Erosion Evaluation Program (SWEEP). Primary particle size analysis indicated no statistical difference in the size of primary soil particles among tillage or residue treatments. Aggregate size distribution analysis showed that tillage affected the freely-available PM10 content of the soil in the field. No tillage resulted in the lowest freely-available PM10 content as compared with other tillage treatments. Simulations made with SWEEP suggest that PM10 emissions and soil loss are potentially highest for conventional tillage and lowest for no tillage while emissions and soil loss were lower from spring disk than autumn chisel plow. Tillage practices can therefore affect air quality and long range transport of fine particulates emitted into the atmosphere during high wind events in subarctic Alaska.

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1. Introduction

Wind erosion is a serious problem in many areas of the United States, including Interior Alaska where the predominant land use type is forest. Agricultural enterprises in Interior Alaska were limited to gardens and small hectarage until the early 1980s when 50,000 hectares of indigenous forest were cleared for small grain production. Soils suitable for agricultural crop production cover 7 million hectares and are largely restricted to flood plains and terraces and outwash plains in the lowlands of Interior Alaska and in broad valleys and rolling uplands of Kushokwim Highlands and highlands of Interior Alaska (Rieger et al., 1979). Schoephorster (1973) found that agricultural soils in Alaska are susceptible to wind erosion. These soils are often shallow and form very slowly due to low annual temperatures, thus management practices to conserve the soil are particularly relevant to sustainable agriculture in the region. Lyles (1983) computed the potential wind erosion using wind-energy distributions and reported that 43% of erosive wind energy occurs during the periods of January through March in Big Delta, Alaska. Pierson et al. (1988) also reported that surface winds contribute to wind erosion in the Alaska subarctic region.

Wind erosion not only affects the sustainability of the soil resource base and therefore the agricultural enterprise, but also the quality of other natural resources. For example, wind erosion can greatly increase the frequency and amount of dust in the air (Prospero et al., 1983; Tegen and Fung, 1995; Zhibao et al., 2000). Airborne dust can cause regional haze and affect the aesthetic quality of our national monuments and parks and wilderness areas. Airborne dust can also lead to exceedance of air quality

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standards (Sharratt and Lauer, 2006), namely PM10 guidelines set by the United Nations and many national governing organizations. Wind erosion can cause emission of PM10 and subsequently lead to societal health concerns, including skin irritations and diseases, eye irritations, shortness of breath, and respiratory disorders (Clausnitzer and Singer, 1996). Long range transport of the dust generated by wind erosion processes has become a global problem and generated interest in Europe (Bègue et al., 2012; Goossens et al., 2001), Africa (Bielders et al., 2000), Asia (Han et al., 2008; Zhibao et al., 2000), Australia (Gillieson et al., 1996; McGowan and Clark, 2008), and South America (Buschiazzo et al., 1999; Gasso et al., 2010). Lin et al. (2007) revealed that the contribution of long-range transport of dust to elevated PM10 concentrations in northern Taiwan province of China during winter and spring to be in the range of 50–75% when strong northeasterly winds prevailed. Betzer et al. (1988) also suggested that the content of PM50 and PM60 (particulate matter respectively ≤50 and 60 μm in aerodynamic diameter) has increased in the atmosphere over the North Pacific Ocean due to the long-range transport from eastern Asia.

Tillage and residue management can impact wind erosion and fine particulate emissions. Agricultural operations such as plowing, mowing, cutting, and baling have the potential to increase dust emissions. In California's San Joaquin Valley, Clausnitzer and Singer (1997) found that tillage operations prior to sowing or after harvest accounted for 67% of all farming operations and 82% of the respirable dust while inter-row cultivation and harvest of crops accounted for 33% of all farm operations and only 18% of the respirable dust. Conservation tillage, however, has the potential to reduce wind erosion and dust emissions. Sharratt and Feng (2009) found a 15–65% reduction in soil loss and 30–70% reduction in PM10 loss from agricultural lands managed using undercutter tillage versus conventional tillage during high wind events in eastern Washington. Similarly, Sharratt et al. (2010) found that less invasive tillage operations during the fallow phase of a winter wheat–summer fallow rotation resulted in lower soil and PM10 flux during high winds. Knight and Lewis (1986) reported that aggregates were more stable for no tillage than for more aggressive tillage operations in subarctic Alaska. In addition to tillage, the type of crop grown can also influence soil erodibility. Crop residue biomass, height, orientation and stem diameter and density influence the protection of the soil surface from wind erosion after harvest (Bilbro, 1991). Bilbro and Fryrear (1994) developed equations for predicting the impact of residue cover and plant silhouette area on the soil loss ratio (ratio of sediment loss from a protected soil and bare soil). They found the soil loss ratio to vary as an exponential function of residue cover and stem silhouette area.

Little information is available that describes PM10 and airborne dust emission potentials from agricultural soils in the subarctic region subject to different tillage and residue management practices. The objective of this study was to identify the impact of long-term tillage and residue management on fine soil particulate emissions of agricultural soils in Interior Alaska.

2. Materials and methods

2.1. Tillage and residue practices

A long-term tillage and crop residue experiment was conducted at the University of Alaska-Fairbanks Agriculture and Forestry Experiment Station located near Delta Junction, Alaska (63°56′ N, 145°20′ W) (Fig. 1). Delta Junction has a mean annual air temperature of 3 °C and precipitation of 300 mm. Straw and tillage treatments were initiated in the spring of 1983 on a Volkmar silt loam (coarse-silty over sandy or sandy-skeletal, mixed, non-acid Aquic Cryochrept).

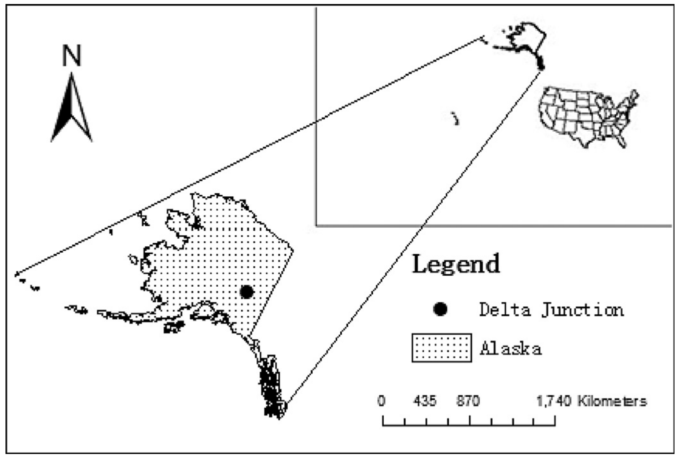

Fig. 1. Location of the field site within Alaska in the USA.

The experimental design was a strip plot with tillage as the main treatment and straw amount as the secondary treatment. The design had three replications with main plots 23 × 120 m and sub-plots 23 × 40 m. Tillage treatments consisted of (1) CT: conventional tillage with one disking both in the autumn, after harvest, and in the spring, prior to sowing of barley; (2) DO: minimum tillage with a single disking to a depth of 0.1 m in the spring prior to sowing barley; (3) CP: minimum tillage with chisel plowing to a depth of 0.1 m in the autumn using 0.08 m twisted points followed by sowing of barley in the spring; and (4) NT: no-tillage with the soil being disturbed only during the sowing operation in spring. Straw treatments consisted of (1) SS: stubble and loose straw remaining on the soil surface following harvest of barley and (2) NSS: no stubble or loose straw on the soil surface with stubble and straw raked, baled, and removed after harvest of barley.

2.2. Soil sampling

Soil properties and surface characteristics affecting wind erosion of tillage and residue treatments were assessed in the spring of 2004 or 20 years after initiating the long-term experiment (Sharratt et al., 2006a,b). Application of tillage treatments amid crop failures in 2002 and 2003 (Sharratt et al., 2006a) resulted in little addition of barley residue to the soil and thus low residue biomass and cover for CT, DO, and CP treatments in 2004 (Table 1).

Although Sharratt et al. (2006a) measured the dry aggregate size distribution, they did not examine the fine particle composition of the soil subject to various management practices. The fine particle composition of the soil was determined from samples collected immediately after sowing barley in each tillage and straw treatment in the spring of 2004. Soil samples were taken at 10 locations between wheel tracks and crop rows in each plot using a flat-bottom shovel to measure PM10 and PM53 (particulate matter ≤53 μm in diameter) fraction. The PM53 fraction was of interest due to susceptibility of this fraction to long range transport (Betzer et al., 1988). Soil samples, approximately 1 kg obtained from the upper 20 mm of the soil profile, were placed on a plastic tray and air dried inside a regulated temperature (25 °C) facility. Before sieving the soil, straw was carefully removed that remained in the sample. The sample was hand sieved through screens having nominal openings of 12.5, 6.4, and 2.0 mm and then mechanically sieved through a nest of sieves having nominal openings of 0.85, 0.42, 0.25, 0.125, and 0.053 mm using a modified sieve shaker

Table 1
Input parameters for conventional (CT), spring disk (DO), autumn chisel plow (CP) and no tillage (NT) required by the SWEEP model.

Parameter	Tillage				Source of data
	CT	DO	CP	NT	
Residue average height (m)	0.15	0.15	0.15	0.15	Sharratt et al. (2006a)
Residue stem area index ($m^2 m^{-2}$)	0.001	0.004	0.007	0.22	This study
Residue leaf area index ($m^2 m^{-2}$)	0	0	0	0	Sharratt et al. (2006a)
Residue flat cover ($m^2 m^{-2}$)	0.022	0.03	0.04	0.998	Sharratt et al. (2006a)
Growing crop average height (m)	0	0	0	0	Sharratt et al. (2006a)
Growing crop stem area index leaf area index ($m^2 m^{-2}$)	0	0	0	0	Sharratt et al. (2006a)
Growing crop leaf area index ($m^2 m^{-2}$)	0	0	0	0	Sharratt et al. (2006a)
Rowing spacing (m)	0	0	0	0	Sharratt et al. (2006a)
Seed placement (cm)	1	1	1	1	Sharratt et al. (2006a)
Number of soil layers	1	1	1	1	Sharratt et al. (2006a)
Layer thickness (mm)	20	20	20	20	Sharratt et al. (2006a)
Bulk density ($Mg m^{-3}$)	0.8	0.67	0.77	0.71	Sharratt et al. (2006a)
Sand fraction ($Mg\ Mg^{-1}$)	0.29	0.3	0.29	0.34	This study
Very fine sand fraction ($Mg Mg^{-1}$)	0.25	0.25	0.25	0.25	This study
Silt fraction ($Mg Mg^{-1}$)	0.61	0.6	0.61	0.55	This study
Clay fraction ($Mg Mg^{-1}$)	0.099	0.095	0.1	0.098	This study
Rock volume fraction ($m^3 m^{-3}$)	0	0	0	0	This study
Average aggregate density ($Mg m^{-3}$)	1.5	1.5	1.5	1.5	Feng and Sharratt (2009)
Average dry aggregate stability [ln($J kg^{-1}$)]	1.6	1.6	1.6	1.6	Feng and Sharratt (2009)
Geometric mean diameter of aggregate size (mm)	1.7	2.7	3	2.8	Sharratt et al. (2006a)
Geometric standard deviation of aggregate size ($mm mm^{-1}$)	13.37	8.59	15.2	15.35	Sharratt et al. (2006a)
Minimum aggregate size (mm)	0.001	0.001	0001	0.001	Feng and Sharratt (2009)
Maximum aggregate size (mm)	43	43	43	43	Feng and Sharratt (2009)
Soil wilting point water content ($Mg Mg^{-1}$)	0.2	0.2	0.2	0.2	Sharratt (1990)
Surface crust fraction ($m^2 m^{-2} \times 100$)	100	100	100	100	Sharratt et al. (2006a)
Surface crust thickness (mm)	1	1	1	1	Sharratt et al. (2006a)
Loose material on crust ($m^2 m^{-2}$)	0	0	0	0	Sharratt et al. (2006a)
Loose mass on crust ($kg m^{-2}$)	0	0	0	0	Sharratt et al. (2006a)
Crust density ($Mg m^{-3}$)	0.9	0.9	0.9	0.9	Feng and Sharratt (2009)
Crust stability [ln($J kg^{-1}$)]	0.6	0.6	0.6	0.6	Feng and Sharratt (2009)
Allmaras random roughness (mm)	6	9.5	11.7	5.4	Sharratt et al. (2006a)
Ridge height (mm)	0	0	0	0	Sharratt et al. (2006a)
Ridge spacing (mm)	0	0	0	0	Sharratt et al. (2006a)
Ridge width (mm)	0	0	0	0	Sharratt et al. (2006a)
Ridge orientation (deg)	0	0	0	0	Sharratt et al. (2006a)
Dike spacing (mm)	0	0	0	0	Sharratt et al. (2006a)
Snow depth (mm)	0	0	0	0	Sharratt et al. (2006a)
Hourly surface water content ($m^3 m^{-3}$)	0.06	0.06	0.05	0.23	Sharratt et al. (2006a)

(Gilson model SS-45A); the shaker allowed three dimensional sieving and was operated at a frequency of 2 Hz for 60 s. The PM53 size fraction was further processed to obtain PM10 content using a sonic sieve apparatus (Advantech Manufacturing, New Berlin, WI). The apparatus was adjusted prior to analysis to obtain reproducible results and to ensure that <1% of the material remaining on the screen was of a smaller size than the screen opening after an additional 1 min of sieving of the silt loam (ASTM, 2003).

Primary particle size distributions were measured by the Mastersizer laser diffraction analyzer (Malvern Instruments Ltd.). Samples were pretreated before analysis using sodium acetate to dissolve carbonates and hydrogen peroxide to oxidize organic matter. Samples were rinsed with deionized water, centrifuged, and excess supernatant was decanted. Each sample was dispersed with sodium hexametaphosphate by agitation for 16 h and analyzed in a deionized water suspension with no sonication.

The soil comprised of aggregates and primary particles ≤53- and 10-μm in diameter was determined from the two size fractions; percent mass ≤53 and 10 μm was assessed based upon the mass passing through the respective sieve and the total weight of the soil sample.

2.3. Single-event Wind Erosion Evaluation Program (SWEEP)

The SWEEP was used to assess potential PM10 emissions and soil loss from tillage and residue management systems. SWEEP is a process-based model that simulates wind erosion and PM10 emissions from agricultural lands for a high wind event occurring on a single day. The model consists of the erosion submodel of the Wind Erosion Prediction System (WEPS) coupled with a user interface (USDA, 2008). Wind erosion is initiated when friction velocity exceeds the threshold friction velocity of the surface. Friction velocity is determined from the aerodynamic roughness of the surface and the log-law wind speed profile. Threshold friction velocity is defined as the minimum velocity that initiates saltation of aggregates on the surface. Threshold friction velocity is determined from soil aggregate parameters (geometric mean diameter and geometric standard deviation, minimum and maximum aggregate size, and aggregate density) and surface conditions including clod/crust cover, loose material on crust, surface roughness, flat biomass cover, surface soil water content, and soil wilting point water content.

SWEEP requires information about the crop, soil properties, surface characteristics, and weather. Parameters required by SWEEP are listed in Table 1. Most parameters were measured during the spring field campaign in 2004. Parameters specific to aggregate density and stability, crust density and stability, and soil wilting point water content were available from the literature. The source of information regarding each parameter is provided in Table 1. Barley had not emerged at the time of our study, thus there was no growing crop (growing crop parameters were set to zero). Crop residue was apparent on the soil surface after sowing, thus residue flat cover was measured at each of the 10 locations in the plot (Sharratt et al., 2006a). Residue stem area index (SAI) was estimated from standing residue biomass (Sharratt et al., 2006a) and the relationship between SAI and standing residue biomass for wheat (Sharratt et al., 2012). Soil properties were measured at 10 locations in the plot (Sharratt et al., 2006a) or were obtained from Sharratt (1990) or Feng and Sharratt (2009).

Wind erosion is most important during spring in Alaska because soils are driest and most exposed at this time of year, especially around the time of sowing. Spring tillage is performed in late April or early May and the soil is sown to barley about mid-May thus the soil is most exposed to winds during May. Wind speeds are the highest in winter and then decline through the summer months with wind speeds in May being higher than wind speeds in subsequent summer months (Shulski and Wendler, 2007). Soils begin to freeze in September and are snow covered for the winter beginning in October and ending in April. Thus, wind erosion typically does not occur in autumn (wet season), winter (snow cover) or summer (plant cover). Wind speed data during May from 2000 to 2010 were obtained from the National Climatic Data Center. Hourly wind speed data were obtained for the Big Delta weather station (located at the Allen Army Airfield in Delta Junction and about 19 km west-north-west of our experimental site) to simulate the soil loss and PM10 emission potential in May from 2000 to 2010.

2.4. Statistical analysis

Experimental field data were analyzed as a strip plot design using analysis of variance in SAS (SAS Institute, Cary, NC) to test for differences among treatments. Means were considered significantly different at $P < 0.05$ and least significant difference (LSD) was used to separate treatment effects and interactions.

3. Results and discussion

Primary particle composition of the soil for our tillage treatments is indicated in Fig. 2. The content of sand, silt, and clay ranged from respectively 29.0 to 34.4%, 55.8 to 61.8%, and 9.8 to 10.0% across tillage treatments. Tillage practices did not affect soil composition since the tillage treatments were located on uniform terrain.

Primary particle size analysis indicated that the D10 (10th percentile of the size distribution) of the different tillage practices ranged from 2.3 to 2.8 μm. In addition, the D50 ranged from 29.7 to 35.8 μm and the D90 ranged from 119.5 to 172.9 μm (Table 2). Although the range was large in particle size for these three percentiles, there was no statistical difference in the size of particles among tillage treatments. Our results are similar to Chang and Lindwall (1989) who found there was no difference in particle size distribution with tillage practices in Alberta, Canada.

Table 2
The 10, 50, and 90th percentile (D10, D50, and D90, respectively) of the primary particle size distribution for conventional tillage (CT), spring disk (DO), autumn chisel plow (CP) and no tillage (NT).

Tillage	D10 (μm)	D50 (μm)	D90 (μm)
CT	2.32 ± 0.23[a]	31.02 ± 1.19	122.72 ± 9.30
DO	2.38 ± 1.01	31.74 ± 2.09	119.54 ± 19.08
CP	2.32 ± 0.38	29.71 ± 2.09	134.83 ± 24.95
NT	2.85 ± 2.10	35.81 ± 4.36	172.89 ± 49.07

[a] Values after the ± symbol represent the standard error.

The PM10 content obtained from the primary soil particle size distribution ranged from 22.2 to 25.4% with a mean value was 23.7% to mean value of 23.7% (Fig. 3). Although this large percentage reveals a high emission potential of PM10, there was no difference in PM10 content among tillage treatments. Li et al. (2013) reported that the PM10 content of soils along the Tarim River in China ranged from 4 to 65%, which was the major fraction of the windblown sediment. The results from the dispersed particle size analysis revealed that the content of PM53 varied from 65.6 to 71.0% (Fig. 3) across the tillage treatments. Although PM53 is very susceptible to long range transport in the atmosphere and contributes to poor air quality as a result of containing significant amounts of soil nutrients (Zobeck et al., 1989) and contaminants (Pye, 1987), no differences were found among tillage treatments.

The freely-available PM10 and PM53 content in the field, obtained from dry sieving soil collected in the field to obtain the aggregate size distribution, ranged from 0.011 to 0.102% and 1.0 to 9.4%, respectively, across straw and tillage treatments. Straw management appears to have some effect on the PM10 content for the CT and DO treatments, while the fraction of PM53 was influenced by straw management only for the CT treatment

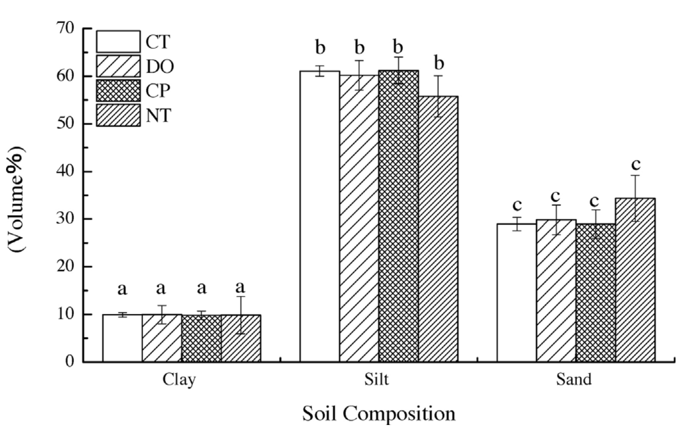

Fig. 2. Soil composition for conventional tillage (CT), spring disk (DO), autumn chisel plow (CP) and no tillage (NT). Means labeled with the same letter were not different at $P = 0.05$ as determined by analysis of variance.

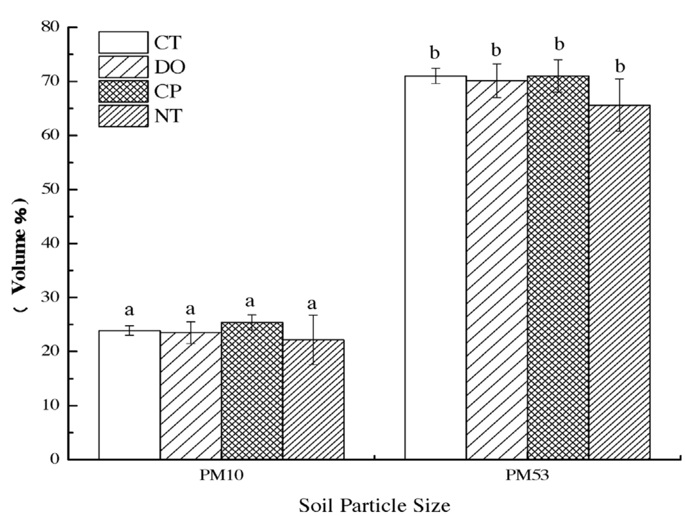

Fig. 3. Percentage of PM10 and PM53 obtained from the soil primary particle size distribution for conventional tillage (CT), spring disk (DO), autumn chisel plow (CP) and no tillage (NT). Means labeled with the same letter were not different at $P = 0.05$ as determined by analysis of variance.

Table 3
Percentage of PM10 and PM53 obtained from the aggregate size distribution of conventional tillage (CT), spring disk (DO), autumn chisel plow (CP) and no tillage (NT). Straw treatments included retaining (SS) and removing (NSS) stubble and straw from the soil after harvest of barley.

Tillage treatment	Straw treatment	PM10 (%)	PM53 (%)
CT	NSS	0.065	8.102
	SS	0.102	9.419
CP	NSS	0.067	8.284
	SS	0.086	7.506
DO	NSS	0.055	4.762
	SS	0.089	5.428
NT	NSS	0.013	1.372
	SS	0.011	0.972
LSD (0.05)		0.025	1.302

(Table 3). For these tillage treatments, retaining stubble and straw on the soil surface resulted in a higher PM10 and PM53 content than removing stubble and straw from the soil after harvest. Straw and stubble remaining on the soil surface not only offer physical protection to the soil, but are also important in reducing wind erosion by enhancing soil stability (Black, 1973; Smika and Greb, 1975) and surface roughness. Tillage affected the PM10 and PM53 content of the field soil with NT resulting in the lowest PM10 and PM53 content as compared with other tillage treatments (Table 3). The low content of PM10 and PM53 in NT likely resulted from biophysical conditions that promoted the formation of larger aggregates in NT compared to other tillage treatments in this study (Sharratt et al., 2006a). Although PM10 content did not vary among the CT, DO and CP treatments, PM53 content was lower for DO than CT and CP (Table 3).

Tillage did not affect the primary particle size distribution, but did influence the aggregate size distribution. The difference between PM10 or PM53 content obtained by particle size analysis (Fig. 3) and sieving dry soil collected in the field (Table 3) is indicative of the reduction in PM10 or PM53 content associated with aggregation of primary soil particles $\leq$10 or 53 μm in size caused by natural processes and agricultural management. The reduction in PM10 or PM53 content associated with aggregation could only be determined in this study by assuming the volumetric size distribution of primary particles, as measured by the Mastersizer, was equivalent to the mass size distribution of primary particles. This assumption is valid when particle density is independent of particle size (Tan, 2000), which we assume for soil particles. Based upon this equivalency, the reduction in PM10 content caused by aggregation for the NSS treatment was 99.7, 99.8, 99.7, and 99.9% for CT, DO, CP, and NT. For the SS treatment, the reduction in PM10 content caused by aggregation was 99.6, 99.6, 99.7, and 99.9% for CT, DO, CP and NT. Similar to PM10, there was a reduction in PM53 content associated with aggregation of primary soil particles. For the NSS treatment, the reduction in PM53 content caused by aggregation was 88.7, 93.2, 88.3, and 97.9% for CT, DO, CP, and NT. For the SS treatment, the reduction in PM53 content caused by aggregation was 86.7, 92.2, 89.4, and 98.5% for CT, DO, CP and NT. These results suggest that tillage influenced soil aggregation with NT enhancing the formation of aggregates >10 and 53 μm in size as compared with other tillage treatments. The lower percentage of PM10 and PM53 in the NT treatment (Table 3) suggests that direct emission of PM10 and PM53 will be potentially lower from NT than from other tillage treatments.

High wind events (days with sustained wind speeds in excess of 6.4 m s^{-1} at a height of 3 m) occurred at least once during the month of May every year over the decadal period of 2000–2010 (Table 4). Simulated loss of PM10 and soil, based upon crop and soil parameters specific to each tillage practice (Table 1), during these May high wind events are indicated in Fig. 4. The NT treatment produced no erosion while the CT, CP, and DO treatments resulted in simulated soil and PM10 loss across these events. The simulated monthly loss of PM10 during these high wind events ranged from 0.022 to 0.065 kg m^{-2}, 0.016 to 0.049 kg m^{-2}, and 0.004 to 0.012 kg m^{-2} for the CT, CP, and DO treatments, respectively. Likewise, the simulated monthly loss of soil ranged from 1.086 to 3.258 kg m^{-2}, 0.892 to 2.648 kg m^{-2}, and 0.816 to 2.448 kg m^{-2} for the CT, CP, and DO treatments, respectively. These ranges in PM10 and soil loss appear to be comparable or larger than loss simulated in other regions of the world. For example, Feng and Sharratt (2007) used SWEEP to simulate PM10 and soil loss from a loessial soil managed in a conventional winter wheat–summer fallow rotation within the Columbia Plateau of the United States and found that PM10 loss ranged from 0 to 0.01 kg m^{-2} and soil loss from 0 to 0.3 kg m^{-2} across six high wind events. Hagen (2004) reported a range in simulated soil loss from 0 to 4.83 kg m^{-2} using SWEEP for single high wind events across 7 locations in the United States. Although Van Donk and Skidmore (2003) found no simulated erosion from an established winter wheat field (height of wheat plants ranged from 0.04 to 0.15 m) using SWEEP for two high wind events in Colorado, they reported simulated soil loss of 1.07 and 4.43 kg m^{-2} for these same events but for soils that were dry and devoid of vegetation. Funk et al. (2002) simulated a soil loss of 0.11–10.46 kg m^{-2} from a sandy soil using SWEEP across 21 single high wind events in Germany.

Straw treatments influenced the residue cover and standing residue biomass (Sharratt et al., 2006a), which are important in simulating PM10 and soil loss with SWEEP. Standing residue biomass can be used to assess stem area index (SAI), an important parameter that affects friction velocity. Stem area index and residue cover for straw and tillage treatments are reported in Table 5 and used in

Table 4
High wind events during the month of May from 2000 to 2010.

Year	Date of high wind event	Maximum wind speed (m s^{-1})	Duration of event (h)
2000	15	10.8	13
2001	2	9.7	8
2002	1	12.8	15
2003	9–10	22.6	38
2004	8, 16	9.7, 13.8	13, 17
2005	1, 5	13.9, 11.2	11, 6
2006	4, 5, 20	13.1, 14.8, 13.1	15, 18, 17
2007	23, 31	13.1, 13.1	23, 22
2008	12–13, 15	14.8, 16.8	42, 22
2009	2	9.4	3
2010	11	17.5	22

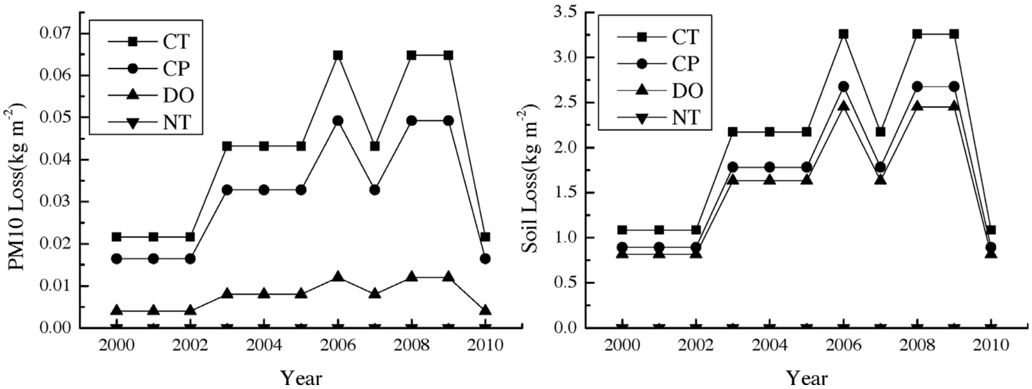

Fig. 4. Loss of PM10 and soil for conventional (CT), spring disk (DO), autumn chisel plow (CP), and no tillage (NT) simulated by SWEEP during the month of May from 2000 to 2010.

SWEEP to assess the influence of straw treatments on PM10 and soil loss. Differences in SAI and residue cover between straw treatments influenced PM10 and soil loss. For example, SWEEP simulated no PM10 or soil loss for the SS treatment during the observed high wind events. For the NSS treatment, no erosion was simulated for the high wind event in 2000. For the remaining years, however, monthly PM10 and soil loss respectively ranged from 0.0256 to 0.0768 kg m^{-2} and 1.896 to 5.688 kg m^{-2}.

Variation in crop and soil parameters due to climate can impact wind erosion. For example, simulated PM10 and soil losses reported in this study (Fig. 4) are based upon the unusual occurrence of precipitation events that resulted in the formation of a soil crust within hours after sowing spring barley. In simulated PM10 and soil losses from a crusted and non-crusted soil during May from 2000 to 2010, we found that losses would be 300% higher for all tillage treatments, except no tillage, when no crust was present on the soil surface. No tillage resulted in zero soil or PM10 loss with or without a soil crust. In addition, Merrill et al. (1999) found that the erodible fraction of soil managed in a wheat–fallow rotation was higher and residue cover and SAI were lower in dry than wet years in Montana. They estimated that these differences in crop and soil parameters contributed to an 11–6100 times higher soil loss in drought years. Although our simulated results are based upon crop and soil parameters measured after sowing barley in 2004, we examined the impact of variations in residue cover and SAI on PM10 and soil loss across years. Our analysis is restricted to variations in these crop parameters because limited agronomic and soil data have been collected on a regular basis since the inception of the long-term study. Annual continuous barley grain yield from the experimental treatments have been found to range from about 500 to 3500 kg ha^{-1} (Sharratt, 1998). From this range in yield, Sharratt et al. (2006a) estimated total residue biomass would vary from 100 to 700 kg ha^{-1} for CT, 170 to 1200 kg ha^{-1} for DO, 200 to 1350 kg ha^{-1} for CP, and 300 to 1950 kg ha^{-1} for NT while residue cover would vary from 0.05 to 0.3 $m^2\,m^{-2}$ for CT, 0.1 to 0.5 $m^2\,m^{-2}$ for DO and CP, and 0.2 to 0.7 $m^2\,m^{-2}$ for NT after sowing barley in spring. Standing residue biomass composes less of the total residue biomass on the soil surface with time after harvest due to decomposition, but we assumed standing biomass to represent about 30% of total biomass in spring (Sharratt et al., 2006a) even though this percentage has been found to vary from 15% (Van Donk et al., 2008) to 40% (Veseth, 1985). From standing biomass, the SAI was estimated to vary from 0.004 to 0.029 $m^2\,m^{-2}$ for CT, 0.007 to 0.050 $m^2\,m^{-2}$ for DO, 0.008 to 0.056 $m^2\,m^{-2}$ for CP, and 0.013 to 0.081 $m^2\,m^{-2}$ for NT. We simulated PM10 and soil loss based upon 1) minimum residue cover and SAI and 2) maximum residue cover and SAI expected based upon the range in grain yield reported by Sharratt (1998) for the tillage treatments. These simulations assumed no changes in crop and soil parameters listed in Table 1 other than the soil was not crusted. Our results suggest that both PM10 and soil loss will be influenced by changes in residue cover and SAI in all tillage treatments except no tillage. Simulations based upon expected minimum or maximum values of residue cover and SAI resulted in zero PM10 and soil loss for no-tillage. For CT, DO, and CP treatments, PM10 or soil loss differed by 400% when simulating erosion using minimum and maximum values of residue cover and SAI. These simulations suggest a 400% higher PM10 and soil loss following years when environmental conditions (e.g., drought) cause low grain yields and residue biomass production as compared to years when conditions (optimal moisture) cause high grain yield and biomass production.

The simulated results showed that CT had the highest PM10 and soil loss as compared with other tillage treatments and the NSS treatment had a higher PM10 and soil loss as compared with the SS treatment during the observed events. Sharratt et al. (2010) suggested PM10 flux were typically lower for reduced or conservation tillage than for conventional tillage. Singh et al. (2012) reported that atmospheric PM10 concentrations within 0.3 m of the soil surface were significantly higher for conventional tillage compared to conservation tillage and no tillage during high wind events in the Columbia Plateau region of the Pacific Northwest United States. Our simulations suggest that conservation tillage practices (DO, CP, and NT) reduce the wind erosion and fine particulate emission potential of subarctic agricultural soils.

Table 5
Stem area index and residue cover of conventional (CT), spring disk (DO), autumn chisel plow (CP), and no tillage (NT) with no stubble and straw (NSS) and stubble and straw (SS) remaining on the soil surface after harvest of barley. These parameters were used in SWEEP to assess PM10 and soil loss of straw treatments.

Parameter	NSS				SS			
	CT	CP	DO	NT	CT	CP	DO	NT
Stem area index ($m^2\,m^{-2}$)	0.0017	0.0049	0.0036	0.181	0.0017	0.009	0.0054	0.2363
Residue cover (%)	1.42	3.67	2.5	99.92	2.92	4.48	3.58	99.58

4. Conclusions

Tillage and residue management have the potential to minimize wind erosion and emission of fine particles in subarctic Alaska. Primary particle size distribution analysis revealed that tillage practices did not affect soil composition since the tillage treatments were located on uniform terrain. Aggregate size distribution analysis, however, indicated that NT promoted aggregation of fine soil particles and resulted in the lowest freely-available PM10 and PM53 content in the field as compared with other tillage treatments. Simulations with SWEEP also showed that PM10 emissions would be greater for CT, thus conservation practices play an important role in minimizing soil erosion and improving air quality in subarctic regions.

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